Fire Use Over a Southwestern Elevational Gradient: Effects of 2003 Fires

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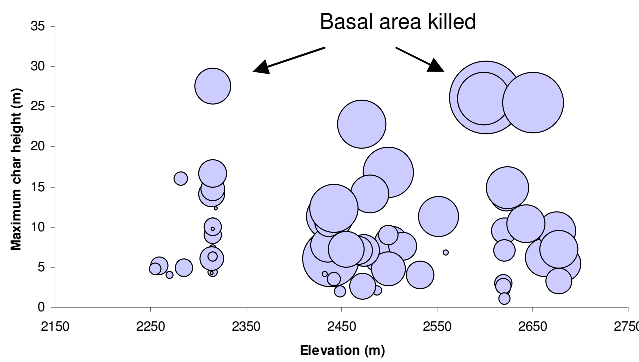

Wildland fires managed for resource benefits on the North Rim, Grand Canyon National Park, in 2003. Upper right: variability in fire intensity and basal area killed increased with elevation. Lower right: low-intensity backing fire on Powell Plateau. Left: burning in mixed conifer forest, Rose Fire. Pictures courtesy National Park Service.

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Abstract

In 2003, a series of wildland fires managed for resource benefits burned over approximately 7,865 ha (19,426 acres) of forests across an elevational gradient in Grand Canyon National Park, Arizona. The fires burned 82 permanent pre-established monitoring plots. We remeasured the burned plots and 43 additional plots on unburned companion sites and compared burned and unburned sites before and after the fires. Fire intensity (measured by maximum bole char) and severity (basal area mortality) increased in mean and variability with increasing elevation. At the low-elevation ponderosa pine-Gambel oak site, tree density declined significantly in the burned area, but basal area was unchanged. At the middle-elevation mixed-conifer site (which did not have an unburned companion) and the high-elevation spruce/fir/aspen site, both density and basal area declined. The thinning effect of fire was concentrated on smaller, shorter, and fire-susceptible trees. Small-diameter trees (< 20 cm diameter at breast height) made up 79%-95% of all tree mortality. Shade-tolerant coniferous species favoring mesic conditions, especially true firs and spruce, experienced disproportionate mortality (31%-82% basal area decline), while fire-resistant ponderosa pine and Douglas-fir tended to survive (2%-8% basal area decline). Delayed mortality, trees on the burned sites that died between 2004, the first growing season following the fires, and 2005 accounted for only 4.2% of trees dying at the low site, but 15.6% and 11.2% at the middle and high sites, respectively. Regeneration density and forest floor depth and woody debris generally declined more sharply in burned than unburned areas, but patterns were more variable than the changes in overstory tree structure. Even though an unusually long fire-free period (1880-2003) had occurred at the middle- and high-elevation burned sites, fire effects were consistent with restoration of historical patterns, moving the ecosystems closer toward historical reference conditions while simultaneously reducing the living, dead, and ladder fuels that made the forest vulnerable to uncharacteristically severe fire. These changes help make the forests more resistant to the expected increases in fire size and severity under future climate conditions.

Keywords

Ponderosa pine, Gambel oak, white fir, Douglas-fir, spruce, aspen, Kaibab Plateau, fuels.

Introduction

Recognition that fire plays an important ecological role in southwestern forests goes back over three-quarters of a century (Leopold 1924), but accomplishing even a limited reintroduction of fire in practice has proven to be challenging (Parsons 2000, Miller and Parsons 2004). The federal policy of wildland fire use (WFU) for resource benefits, applied by Federal resource management agencies since 1995, is designed to give managers flexibility in responding to naturally ignited wildfires (Zimmerman and Bunnell 2000, Miller 2003, Stephens and Ruth 2005). Under an approved fire management plan, wildland fires that meet pre-defined criteria can be used to accomplish goals such as fuel reduction, re-introduction of historical patterns of fire behavior, and restoration of fire-maintained patterns of species composition or forest structure (Zimmerman and Bunnell 2000, Dale 2006).

The difference in fire regime characteristics between current fire patterns and historical fire patterns has been a central theme in assessing the impact of wildland changes associated with fire exclusion and other human-caused disturbances. Examples include the nationwide program of coarse-scale fuel mapping (Schmidt et al. 2002), applied in developing the Fire Regime Condition Class assessment (FRCC, Schmidt et al. 2002), and a finer-scale approach used in the Sierra Nevada of California, called the Fire Regime Interval Departure (FRID, Caprio and Graber 2000). These approaches rely on a quantitative comparison of modern fire recurrence vs. a reference condition of historical fire frequency prior to European settlement to provide a spatially explicit assessment of the departure from historical fire regimes. Areas with high departure, especially those where fires historically burned mostly with low severity, are considered to be at high risk of "losing key ecosystem components" (Schmidt et al. 2002:8) and a variety of restorative actions, not just re-introduction of fire, may be required to regain historical characteristics. Hann et al. (2003) sought to develop a improved "disturbance departure and fragmentation index" by incorporating a variety of ecologically important variables modeled across scales in addition to fire frequency.

A key problem with assessments based on fire frequency comparisons is that the great majority of wildlands appear to have dim prospects for regaining anything close to historical fire frequency under modern constraints (Brown et al. 1994). In a nationwide review, Parsons (2000) noted that even for the agencies most disposed to use fire (National Park Service, Forest Service), even in wilderness settings where fire use may be least constrained, and even in ecosystems where fire might be most easily managed because of historical low- or mixed-severity fire regimes, the application of WFU or the previous policy of "prescribed natural fire" was orders of magnitude below a level that would emulate historical fire regimes. A site-specific study by Rollins et al. (2001) calculated modern fire rotations of > 100-200 years based on approximately two decades of wildfire use in the Selway-Bitterroot (Montana) and Gila-Aldo Leopold (New Mexico) Wilderness Complexes. Miller and Parsons (2004) found that fragmentation of wildlands limited the importation of fires from distant ignitions, further reducing even the theoretical possibility of fully restoring natural fire.

Recognizing that supplementing natural ignitions with prescribed fires and/or mechanical treatments raised a new set of concerns about ecological integrity, Parsons (2000) reiterated the dilemma of the difficult choices between the consequences of continued minimal fire use vs. the potentially problematic social and ecological consequences of active management intervention. A third option, of course, is to try to increase fire use, perhaps through demonstrating practical advantages of cost savings in addition to ecological benefits (Dale et al. 2005).

We suggest that there may be another useful option: the actual effects of specific wildland fires may, in some circumstances, be more important than the frequency of fire recurrence. If fires that differed in spacing, season, intensity, or other ways from historical patterns nonetheless conserved or restored ecosystem structures and processes within a natural range of variability, then perhaps long-term management goals might be achievable even under altered modern fire regimes. For example, Miller and Urban (2000) used simulation modeling to suggest that relatively severe prescribed fires could restore mixed-conifer forests in the Sierra Nevada by thinning dense stands of young, fire-susceptible trees.

In southwestern ponderosa pine and mixed-conifer forests, considered among the most negatively impacted by fire exclusion in North America (Allen et al. 2002, Schmidt et al. 2002), there are several lines of evidence to support the hypothesis that fire effects can be more important than fire frequency. First, frequent fires by themselves do not necessarily restore historical characteristics. Three decades of controlled experimentation with re-introduction of frequent surface fire in ponderosa pine have shown remarkably little effect in terms of restoring historical forest structure, composition, productivity, or nutrient cycling (Peterson et al. 1994, Sackett and Haase 1996, Hart et al. 2005), leading people to look for ecological restoration alternatives that include tree thinning and fuel treatments in addition to fire (Covington et al. 1997). Second, in never-harvested pine-oak forests in Arizona and northern Mexico that have continued to experience surface fires after regional fire exclusion, even at much wider spacing than historical fires, attributes of open forest structure, low fuels, and a productive understory community have been maintained, in contrast to paired forests that had complete fire exclusion (Fulé and Covington 1997, Fulé et al. 2002, Gildar et al. 2004, Stephens and Fulé 2005). Third, post-fire studies of relict forest sites showed that these ecosystems were resilient to disturbance (Laughlin et al. 2004), in contrast to forests where a history of logging and grazing had left a legacy of depauperate native plants and numerous invasive exotic propagules (Crawford et al. 2001, Dodge 2005). Even where fire was uncharacteristically intense, to the point of requiring expensive suppression efforts, the effects on forest structure were consistent with restoration in terms of preferentially thinning young and fire-susceptible trees (Fulé et al. 2004) while fostering native understory recovery (Huisinga et al. 2005).

An opportunity to test the effects of fire use on a landscape scale over an elevation gradient appeared after extensive WFU fires in 2003 on the North Rim of Grand Canyon National Park. The Park conserves the largest area of unharvested forest in Arizona, over 48,800 ha (Warren et al. 1982), and has been a leading agency in fire restoration under the prescribed natural fire and WFU policies since the 1980s. Beginning in 1997, we had established a series of permanent plots on the North Rim from ponderosa pine to spruce-fir and aspen forests. We measured tree structure, age, plant community, fuels, fire regime, and reconstructed reference conditions of forest structure and canopy fuels (Fulé et al. 2002, 2003a, 2003b, 2004, Laughlin et al. 2005). The 2003 WFU fires burned over 82 permanent plots, representing 1,872 ha (4,624 acres) of forests. We returned to these plots and companion unburned sites after the fires to ask: (1) how did WFU fires vary in intensity and severity over the elevational gradient? (2) Were tree structural changes (density, basal area, canopy cover, regeneration) consistent with conservation or restoration of historical reference conditions? (3) How did forest floor fuels change following WFU fires? And finally (4) what implications can be drawn for future management of the burned sites and adjacent forests?

Methods

Study Area

The study area is the northwestern portion of the North Rim of Grand Canyon National Park (Figure 1). Powell Plateau is part of a designated Wilderness Area and most of the North Rim, while not Congressionally designated, is also managed following Wilderness guidelines. Weather records were taken from the Western Regional Climate Center (www.wrcc.dri.edu, accessed March 17, 2006). Average annual precipitation at the North Rim ranger station (Bright Angel, elevation 2,542 m [8,338 ft]) from 1948-2005 was 65.8 cm (25.5 inches), with an average annual snowfall of 371 cm (146 inches). The decade 1996-2005 averaged 91% of the long-term average precipitation, but 6 of the 10 years were below average and 2002, one year before the WFU fires, was the driest year in the North Rim records (excluding years with missing data) (Figure 2). Temperatures ranged from an average July maximum of 25° C (78° F) to an average January minimum of -8.2° C. North Rim soils are predominantly of the Soldier series, derived from Kaibab limestone (Bennett 1974). Soils[1] at the low-elevation site were Typic Paleustalfs. At the middle and high-elevation sites, soil textures ranged from coarse to fine loams. Valley soils were Cumulic Haplustolls, soils on 15-40% slopes were Oxyaquic Paleustalfs, and flatter upland areas (2-15% slopes) were Cumulic Haplustolls. Forests varied with elevation, from ponderosa pine and Gambel oak at lower elevation through spruce, fir, and aspen at high elevation. Species associations are listed in Table 1 and scientific names are listed in Table 2.

Fire regimes in ponderosa pine and mixed conifer forests were reconstructed from fire-scarred trees on the low- and middle-elevation study sites by Fulé et al. (2003a, 2003b) and nearby by Wolf and Mast (1998). Composite fire return intervals between the early 1700s through 1879 averaged 3.2-5.5 years (including all fires) and 6.4-9.2 years (for fire dates in which 25% or more of the sample trees were scarred (Fulé et al. 2003b). At the higher elevation study site, Fulé et al. (2003a) used a combination of fire-scar data and tree age and species composition to show that these forests had a mixed-severity fire regime, which consisted of a mixture of low-severity surface fires on warm, south-facing slopes (31-year mean fire return interval) and infrequent, patchy, stand-replacing fires elsewhere (currently 120-230 years since the last fire) (Fulé et al. 2003a). Fires ceased after 1879 across much of the North Rim, probably due

[1] Soil information from unpublished soil survey report, on file at Grand Canyon National Park, Arizona.

to the introduction of livestock, followed by fire suppression policies. The North Rim was fenced to exclude livestock by 1938. No large fires were recorded on fire scars or in Park records at the middle and high-elevation sites after 1879 until the WFU fires in 2003. Both of the low-elevation sites, in contrast, had at least three large, spreading surface fires after 1879 (Fulé et al. 2003b).

Field Methods

From 1997 to 2001, we established permanent plots to measure forest structure and fuels, coupled with collection of dendroecological samples. The plots were established systematically on large landscapes covering an elevational gradient from ~ 2,200 m (7,216 ft) to ~ 2,700 m (8,856 ft). Plot grids were spaced at 300 X 300 m (328 X 328 yards)below 2,600 m (8,528 ft) elevation (Figure 1). Above that elevation, a wider spacing of 600 X 1,200 m (656 X 1,312 yards) was used to measure broader-scale forest variability associated with the mixed-severity fire regime (Fulé et al. 2003a).

Sampling plots, modified from the National Park Service's Fire Monitoring protocol (NPS 2003), were 0.1 ha (0.25 acre) in size, 20 X 50 m (65.6 X 164 ft) and were oriented with the 50-m sides uphill-downhill. All trees were tagged and tree measurements included species, diameter at breast height (dbh), height, crown base height, and tree condition. Trees larger than 15 cm (5.9 inches) dbh were measured on the entire plot; trees between 2.5 and 15 cm (1.0 and 5.9 inches) dbh were measured on a 0.025 ha (0.06 acre) subplot. Trees smaller than 2.5 cm (1.0 inch) dbh (regeneration) were tallied by species and height class on a 50 m^2 (538 ft^2) subplot. Along the 50-m sidelines of the plot, canopy cover was recorded with a vertical densiometer (Ganey & Block 1994); in pre-fire measurements this was measured every 30 cm (11.8 inches) and in 2001-2003, it was measured every 3 m (9.84 ft). Forest floor and woody debris were measured along four 15.24 m (50 ft) planar intersect transects (Brown 1974) located every 10 m (32.8 ft) along the plot centerline. Transect directions were randomly chosen. Litter and duff depths were measured every 1.52 m (5 ft) along each transect, and woody debris was measured by time-lag classes (equivalent to diameter categories; Anderson 1982) of 1 hr, 10 hr, 100 hr, and 1000 hr along each transect. Fuel loadings were calculated from the planar transect data (Brown 1974; Sackett 1980). Dendroecological (tree-ring) data and modeling were applied to reconstruct pre-1880 forest conditions (Fulé et al. 2002) and fire regimes were reconstructed for all the study areas using fire scars and tree age data (Fulé et al. 2003a, 2003b).

Eighty-two plots on the North Rim were burned in the WFU fires of 2003 (Figure 1, Table 1). In 2004, we remeasured all variables on these plots and added measurement of bole char height. Also in 2004, we remeasured 25 unburned plots at the low-elevation comparison site and two unburned plots at the high-elevation comparison site. Post-fire mortality is often not evident in the first growing season following burning (McHugh and Kolb 2003, Fowler and Sieg 2004). Therefore we returned in 2005 to remeasure the condition of all tagged trees on the burned plots. In 2005 we also completed remeasurement of the remaining 16 unburned plots at the high-elevation comparison site.

Statistical Methods

This research is an observational study taking advantage of wildland fires that burned across pre-existing monitoring plots. The design is a before-after, control-impact (BACI) study (Eberhardt and Thomas 1991, Stewart-Oaten et al. 1992) for the low and high-elevation sites where we could measure nearby unburned sites. At the middle-elevation site, only before- and after-fire data could be compared. The scope of inference is limited to these particular sites and fires.

Forest structural variables including tree density, basal area, canopy cover, and regeneration density were compared between burned and unburned sites at low and high elevation with Kruskal-Wallis 2-sample tests. Alpha level was .05. Following a statistically significant result (Mann-Whitney U statistic) for a total variable, such as total basal area, the basal areas of individual species were compared between burned and unburned sites with pairwise Kruskal-Wallis 2-sample tests. Because some species were sparsely distributed at some sites, only species that were present on $\geq 20\%$ of the plots at a given site were tested for statistical significance of changes. Alpha levels for these tests were adjusted by dividing by the number of pairwise tests (Bonferroni correction).

Changes over time from the pre-fire to the post-fire measurements within sites were tested with Wilcoxon signed-ranks tests to take advantage of the repeated measurements on the permanent plots. We followed the same procedure as with the between-site tests: following a statistically significant result (Wilcoxon Z statistic) for a total variable, we proceeded to Bonferroni-corrected pairwise comparisons by individual species.

We used non-metric multi-dimensional scaling (NMS) ordinations (Clarke 1993) to describe changes in tree basal area by species among plots using PC-ORD software (version 4.25; McCune and Mefford 1999). Basal area was selected as the forest structural variable because it is likely to be more reliably reconstructed than past tree density (Fulé et al. 2002, Moore et al. 2004). Three time periods were compared at each elevation: (1) reference conditions reconstructed with dendroecological data (Fulé et al. 2002, 2003a); (2) pre-fire conditions, and (3) post-fire conditions. At the low- and high-elevation sites, both the burned and unburned areas were compared together on the same set of axes. We used the Bray-Curtis distance measure (Faith et al. 1987) with random starting configurations, 40 runs with real data, a maximum of 400 iterations per run, and a stability criterion of 0.00001. A Monte Carlo test with 50 randomizations was used to determine how likely the observed stress value of the final solution would be by chance alone.

Results

Tree Mortality

Across the landscape as a whole, the fires burned with a variety of intensities and effects, but tended to be more intense, as estimated by the maximum bole char height per plot, and more severe in terms of basal area killed as elevation increased (Figure 3). Basal area varied among plots and sites, but the absolute and relative (%) values of basal area killed both serve as useful indicators of severity because they were highly intercorrelated ($r = .93$, $P < .001$). At low elevation, the maximum mortality on any plot reached 13.7 m^2 ha^{-1}, corresponding to 58%. This was the only plot with > 10 m^2 ha^{-1} mortality; the mean mortality was 1.6 m^2 ha^{-1}, corresponding to an average of 6% of pre-fire basal area. In contrast, at the middle-elevation site, 12 of 29 plots had > 10 m^2 ha^{-1} mortality. Mean mortality was 9.5 m^2 ha^{-1}, corresponding to an average of 23%, and reaching a maximum of 32.4 m^2 ha^{-1}. The highest single-plot relative mortality was 83%. Severity was greatest at the high-elevation sites, where 10 of 18 plots exceeded 10 m^2 ha^{-1} mortality. Mean mortality was 14.4 m^2 ha^{-1}, corresponding to an average of 39%, and reaching a maximum of 53.4 m^2 ha^{-1}. Two plots had 100% overstory mortality and a third reached 92% mortality.

Declines in tree density and basal area occurred over all elevations and in both burned and unburned sites, but declines were significantly more pronounced at the middle and high-elevation burned sites. Consistently, density declined more than basal area, burned sites declined more than unburned sites, higher elevations more than lower, and fire-susceptible species more than fire-resistant species (ponderosa pine and Douglas-fir). Changes were minimal at the low-elevation sites, both comparing burned and unburned sites as well as within-site changes (Table 3). Total density and basal area were not significantly different between the burned and unburned sites in the pre-fire measurement. After the fire, density was significantly lower in the burned site ($U = 310$, $P = .04$), but neither pines nor oaks separately were significantly different between burned and unburned sites. Basal area remained not significantly different between sites.

Within the low-elevation sites, the burned site tree density dropped significantly in total ($Z = -3.98$, $P < .001$) and for both ponderosa pine ($Z = -2.65$, $P = .008$) and Gambel oak ($Z = -3.22$, $P = .001$). All New Mexican locust trees, averaging 82 stems ha^{-1} before the fire, were killed or topkilled on the burned site. The unburned site density decline was significant in total ($Z = -2.09$, $P = .04$) and for both ponderosa pine ($Z = -3.17$, $P = .002$) and Gambel oak ($Z = -2.48$, $P = .01$). Basal area in the burned site declined significantly by 6% ($Z = -2.21$, $P = .03$). Ponderosa pine basal area did not change significantly but Gambel oak did ($Z = -3.22$, $P = .001$). Basal area did not change significantly at the unburned site (2% decline).

At middle elevation, since there was no companion unburned site available to compare with the burned site, results are simply the changes over time from the pre-fire to the post-fire measurement. Tree density declined by 54% (Table 3), a significant decrease from pre- to post-fire ($Z = -4.70$, $P < .001$), driven by significant declines in white fir ($Z = -4.37$, $P < .001$), ponderosa pine ($Z = -3.59$, $P < .001$), and aspen ($Z = -4.18$, $P < .001$), but not Douglas-fir. The 23% decline in total basal area was also significant ($Z = -4.29$, $P < .001$), but in this case only white fir and aspen were significantly lower ($Z = -4.23$ and -4.08, respectively, both $P < .001$). Ponderosa pine and Douglas-fir basal areas did not decrease significantly.

The most pronounced declines in tree density and basal area occurred at the high-elevation sites (Table 3). Before the fire, total density and basal area were indistinguishable between the burned and unburned site ($U = 170$ and 168, $P = .80$ and .89, respectively). After the fire, both variables were significantly lower in the burned sites ($U = 79$ and 56, $P = .009$ and $P <$

.001, respectively). Tree density differences were driven by significant differences between sites in subalpine fir and spruce (U = 75 and 62, P = .004 and P = .001, respectively), but white fir, ponderosa pine, aspen, and Douglas-fir were not significantly different. Basal area differences were also due to significant differences between sites only in subalpine fir and spruce (U = 68 and 77, P = .002 and P = .006, respectively).

Within high-elevation sites, the burned site tree density dropped 57%, a significant decrease in total (Z = -3.72, P < .001) and for white fir, spruce, and aspen (Z = -3.07, -2.69, and -3.46, P = .002, .007, and .002, respectively). Basal area declined 42% in the burned site, also significant (Z = -3.72, P < .001), driven by declines in white fir and spruce (Z = -3.01 and -3.46, P = .003 and .002, respectively). In contrast to the findings at low elevation, significant declines also occurred over time in the unburned site. Unburned site tree density decreased significantly by 11%, (Z = -3.72, P < .001), but the only individual species with a statistically significant decline was spruce (Z = -3.12, P = .002). Unburned site basal area declined significantly by 6% (Z = -2.98, P = .003), but no individual species had a statistically significant decline.

Tree mortality in both burned and unburned sites occurred predominantly in the smaller size classes, evidenced both by diameter distributions (Figure 4) and by the disproportionate relationship between the greater reduction in tree density (6%-57%) compared with the lesser reduction in basal area (2%-42%) (Table 3). On the burned sites, the proportion of dying trees ≤ 20 cm dbh was 95% at the low site, 83% at the middle site, and 79% at the high site. On the two unburned sites, 100% of the dying trees at the low site and 74% of the dying trees at the high site were ≤ 20 cm dbh. Delayed mortality, trees on the burned sites that died between 2004, the first growing season following the fires, and 2005 accounted for only 4.2% of trees dying at the low site, but 15.6% and 11.2% at the middle and high sites, respectively (Figure 3). As a proportion of basal area, delayed mortality accounted for < 2%, 9.9%, and 12.2% at the low, middle, and high sites, respectively.

Across all fires, surviving overstory trees (≥ 15 cm dbh) tended to be larger and have lower maximum bole char heights than trees that died (Figure 5, Table 5). These differences were statistically significant for all combinations of species and fires, except for aspen diameter at the high-elevation site and subalpine fir and spruce bole char (Table 5). Species varied in the proportion of large trees dying, with "large" defined as dbh ≥ 37.5 cm (White 1985). Large trees made up 15% of tree mortality for white fir, 29% for subalpine fir, 16% for spruce, 7% for ponderosa pine, 14% for aspen, and none for Douglas-fir.

Shifts in the distribution of basal area by species generally maintained broad variability on forest structure but moved in the direction of historical reference conditions, as illustrated in the non-metric multidimensional scaling ordination diagrams in Figure 6. Note that these diagrams are presented on arbitrary and dimensionless axes, showing relative changes. At all three sites, the distribution of reference plots tended to be somewhat more aggregated than the pre- and post-fire distributions. Arrows in Figure 6 show the change in the geometric centroid from pre- to post-fire patterns. At the low-elevation site, the change was too small to be graphed, consistent with the minimal changes in total basal area (2%-6%). In contrast, the middle-elevation site showed a distinct shift associated with the burn, moving in the direction of the reference pattern. The changes at high elevation were also evident. The centroid of the pre-fire plot distribution—which had become spread more widely than the reference condition—moved sharply back within the cloud of reference plots. However, the centroid of the unburned plot distribution also moved a relatively large distance, about 40% as much as the burned plots.

Canopy Cover

Canopy cover (Table 4) at low-elevation sites did not differ significantly between burned and unburned sites before or after the fire, but declined significantly within both sites over time (burned: $Z = -2.49$, $P = .01$; unburned: $Z = -2.11$, $P = .03$). At middle elevation, canopy cover (Table 4) declined significantly over time ($Z = -2.63$, $P = .009$). At high elevation, canopy cover (Table 4) did not differ significantly between burned and unburned sites before the fire. Post-fire canopy cover between sites and changes over time in the unburned site could not be assessed because of the missing canopy data. However, cover declined significantly in the burned site ($Z = -2.94$, $P = .003$).

Regeneration

Tree regeneration, all trees < 2.5 cm dbh (Table 6), displayed much more variability than did the larger trees. Total regeneration was not significantly different between burned and unburned sites at low and high elevation, either before or after the fires. Within sites, however, only the middle elevation site failed to show significant differences from before to after the fire. Total regeneration increased significantly at the low burned site ($Z = 2.29$, $P = .02$), driven by a significant increase in New Mexico locust ($Z = 2.74$, $P = .006$), which doubled in density. The low unburned site also increased significantly ($Z = 2.29$, $P = .02$), but no individual species had a significant change. At the high site, both the burned and unburned sites declined significantly in regeneration density over time ($Z = -3.42$ and -2.67, $P < .001$ and $P = .007$, respectively), but the only individual species with a statistically significant decrease was aspen, in the unburned site ($Z = -2.79$, $P = .005$). Total regeneration density was not associated with fire severity but was positively correlated with maximum per-plot char height ($r = .25$, $P = .023$).

Forest Fuels

Forest floor depth declined in the burned areas, but relationships between burned and unburned sites were variable (Table 7). Forest floor depth was significantly lower at the unburned low site before the fire ($U = 711$, $P < .001$) but depths did not differ afterwards. The opposite occurred at the high site, where forest floor depth was indistinguishable before burning but was significantly lower in the burned site afterward ($U = 43$, $P < .001$). Within sites, declines in forest floor depth were highly significant at the low, middle, and high-elevation sites ($Z = -5.03$, -4.68, and -3.64, respectively; all $P < .001$). Forest floor depth also declined significantly at the high unburned site ($Z = -2.02$, $P = .04$), but did not change significantly at the low unburned site.

Woody debris biomass was indistinguishable between burned and unburned sites before burning at both low and high elevation, but both fine and coarse woody debris were significantly lower at the low site after the fire (U = 235 and 310, P = .002 and .04, respectively). Coarse woody debris was also significantly lower in the high elevation burned site after the fire (U = 81, P < .01). Within sites, fine woody debris declined significantly at the low, middle, and high-elevation burned sites (Z = -2.44, -4.08, and -2.46, P = .01, < .001, .01, respectively). Fine woody debris increased significantly at the low unburned site (Z = 2.81, P = .005). Coarse woody debris declined significantly at the middle and high-elevation burned sites (Z = -4.70 and -2.72, P = .002 and .006, respectively). There were no significant changes in coarse woody debris biomass from before to after the fires at the unburned sites.

Discussion

Effects of the 2003 fires

Fire variability was correlated with elevation, forest type, and pre-1880 fire regime characteristics. This pattern was similar to the natural pattern of fire behavior on these landscapes prior to recent anthropogenic disruption of fire regimes: moister conditions at higher elevations facilitated the development of mesic, productive forest types that burned less frequently but with greater severity than the xeric, lower-elevation forests (Dutton 1882, Lang and Stewart 1910, Rasmussen 1941, White and Vankat 1984, Wolf and Mast 1998, Fulé et al. 2003a, 2003b). Comparisons with unburned companion sites permitted separation of fire effects from background variability. The fortuitous combination of pre-fire plot locations and fire patterns made it possible to compare burned and unburned sites that bracketed the low- and high-elevation extremes of the fires. We had a sufficiently high N (18 to 36) and were able to take advantage of the power of repeated measurements on the same plots for a statistically reliable evaluation. Mortality on the unburned sites was statistically significant, reflecting the severe drought conditions that dominated most of the measurement period (Breshears et al. 2005). Burned sites had substantially higher mortality, however, and burned-unburned companion sites that were statistically indistinguishable prior to fire became significantly different in tree density and basal area afterwards. The unburned sites also provided a control for the less consistent variables, regeneration density, forest floor, and woody debris. Unlike the tagged tree data, regeneration fluctuated widely and even the planar transect data were highly variable, but these patterns emerged on the unburned sites as well, so they were not confounded with burn effects.

The thinning effect of fire was concentrated on smaller, shorter, and fire-susceptible trees. Since small-diameter trees made up 79%-95% of all tree mortality, diameter distributions were shifted to flatter, less inverse-J patterns, and the potential for fuel ladders was reduced. Large trees also died, however, at rates from 0-7% (Douglas-fir and ponderosa pine, respectively) to 29% (subalpine fir) for trees > 37.5 cm dbh. For comparison, in two recent forest restoration experiments adjacent to the Park over nearly overlapping periods (~ 1998-2005), large ponderosa pine trees died by 5 years post-treatment at rates from 9% to 34% following tree thinning, raking of litter away from the boles of large trees, and prescribed fire

(Fulé et al. 2005, in review). Thus the mortality of large trees of fire-resistant species in the free-burning WFU fires compared favorably to mortality in mechanical + fire restoration treatments.

Fire-susceptible species were the most likely to die, shifting species composition toward fire-resistant species. Shade-tolerant coniferous species favoring mesic conditions, especially true firs and spruce, experienced disproportionate mortality (31%-82% basal area decline), while fire-resistant ponderosa pine and Douglas-fir tended to survive (2%-8% basal area decline). At the high-elevation site, the latter two species made up 26% of total basal area before the fire but 42% afterwards, compared to an unnoticeable shift in the companion unburned site (26% to 28%). Thin-barked deciduous species were also preferentially thinned, with 100% of New Mexico locust stems dying in all three burned sites and 51%-65% declines in aspen basal area.

Regeneration affects the long-term patterns of species composition, however, and the most dense regeneration tended to be fire-susceptible species. Sprouting deciduous species generally dominated regeneration, followed by the fire-susceptible spruce and true firs. Ponderosa pine and Douglas-fir, the most resistant overstory species, were in last place in regeneration density, making up < 1% of all stems. The demographic imbalance does not necessarily forecast future forest structure. Many small sprouts fail to reach the stature of established plants, especially if burned (Gottfried 1980, Harrington 1985). Dense young conifers such as spruce and true firs may be very persistent in the understory due to high shade tolerance, but as shown in the response to the 2003 WFU fires, they are highly susceptible to future thinning. Aspen merits special attention due to the historically important example of the overpopulated Kaibab deer herd, which peaked in the 1920s and left a diminished aspen cohort from the early twentieth century (Rasmussen 1941, Binkley et al. 2006). Current deer populations appear compatible with aspen regeneration, while elk, the species that has the highest impact on aspen, are nearly absent from the Kaibab Plateau (Fulé et al. 2004). Aspen regeneration density declined approximately equally between burned and unburned sites at high elevation, but the absolute values of mean aspen density were still high in the middle- and high-elevation burns, > 2,000 stems ha^{-1}, making up 24% to 59% of the total regeneration at these two burns, respectively. In contrast, high elk populations decimated aspen regeneration within 3 years after a severe fire near Flagstaff, Arizona, approximately 130 km southeast of the present study area (Bailey and Whitham 2002).

Implications

The effects of the 2003 WFU fires should not be overgeneralized for two reasons. First, perhaps more than any other management action, such as prescribed burning or tree thinning, WFU is highly and inherently variable. In relatively moist high-elevation forests, where appropriate conditions to carry fire are encountered less often than in dry lower forests, fire spread won't be feasible unless the fuels and weather are fairly dry. This means that managers must be willing to accept a relatively high risk of intense fire behavior, and that fire effects will be variable in severity from place to place and fire to fire. Patterns detected in the 2003 WFU fires may not necessarily be the same on other fires, even in similar sites. Second, our conclusions about the 2003 WFU fires are necessarily limited to the portions of those fires where pre-existing plots had been located. In the case of the Poplar fire specifically, the plots were located at the northern part of the fire (Figure 1), while areas of relatively more severe burning occurred to the south of the plots (Fire severity maps on file at Grand Canyon National Park, Arizona).

After accepting these caveats, however, the portions of the 2003 WFU fires that we measured can be considered broadly successful in ecological terms. Even though an unusually long fire-free period (1880-2003) had occurred at the middle- and high-elevation burned sites, fire effects were consistent with historical patterns of reduced severity at lower elevation and vice versa. The WFU fire had the least impact at the low-elevation site, where after 1879 the fire regime had shifted from approximately 11 widespread fires per century to fewer than 3, forest structure had not changed a great deal from pre-fire-exclusion conditions (Fulé et al. 2002, Gildar et al. 2004, Figure 6). Forest structural changes were also generally consistent with restoration of a range of variability in reference conditions. The great majority of fire-killed trees were small. Mesic species that had dominated regeneration since fire exclusion were preferentially thinned. Two effects associated with the fires are of concern: the loss of some large and old trees, discussed above, and the increasing presence of cheatgrass in burned areas (Laughlin and Fulé 2005).

The 2003 WFU fires were probably relatively gentle compared to historical fire patterns, except perhaps at the middle-elevation site. At the low elevation sites, though small trees were thinned, fire-caused mortality was barely distinguishable from background mortality in terms of basal area. While most historical fires probably burned at nearly the same season (mid-June onwards), it is likely that the largest areas would have been burned by fast-spreading headfires rather than with the backing fire behavior characteristic of the Powell fire in 2003. The maximum per-plot mortality we observed was 40% of basal area, but previous fires appeared to have killed groups of trees at scales of up to 1-2 ha (Fulé et al. 2003b). At high elevation, the portion of the Poplar fire that we could measure had > 90% mortality on only three out of 18 plots (17%). A reconstruction of the mixed-severity fire regime on this site suggested that as much as 1/3 of the area might have been in early successional stages following patchy stand-replacing burning before fire exclusion (Fulé et al. 2003a).

The relatively low-severity effects of the 2003 WFU fires were probably beneficial in terms of maintaining administrative and public support for burning and helping create favorable logistical conditions for future fires. Now that a large contiguous area of the northwestern North Rim has burned, it may be easier for managers to allow future natural ignitions to burn as WFU fires. Other large fires, notably the Outlet fire (an escaped prescribed burn that covered 5,260 ha in 2000), and large-scale prescribed burns on the Walhalla Pateau have contributed to reduce fuels and foster fire-resistant early successional aspen stands across the North Rim. The east-west boundary line dividing the North Rim from the Kaibab National Forest has long been a source of interagency concern in terms of stopping fires in the heavy fuels of the Park's old forests (Pyne 1989). However, the Poplar fire's location helps to buffer the west side of the boundary. Coupled with older prescribed burn and wildfire sites, the vulnerability of the boundary has decreased substantially since the 1990s. Finally, unlike the sites studied by Miller and Parsons (2004), both rims of Grand Canyon receive large numbers of natural ignitions (Fulé et al. 2002), so the future fire regime should not necessarily be limited by artificial barriers to importing fire. Together with allowing a greater number of wildland fires to burn, it may also be possible to expand the permissible range of fire intensity to more closely match long-term historical patterns, especially at high elevation.

Ecological restoration, whether focused on "restoration of natural fire to wilderness" through WFU (Parsons 2000) or in broader assessments of ecosystem structure, composition, and function compared to historical reference conditions (Allen et al. 2002), can be criticized because the climate conditions of the present and coming decades are unlikely to be like those of the past (Millar and Wolfenden 1999). But the fact that climate is changing is not the point. The issue is not whether future climates will be identical or very similar to those of the past, but rather whether native forest ecosystems can persist under future conditions. Climate change, whether through gradual changes in mean conditions or through increased variability leading to altered severity or frequency of disturbance, may create novel thresholds beyond which a species or ecosystem type cannot survive. But unless or until such a point is reached, the most relevant question for assessing restoration is whether restoration is compatible with long-term ecosystem sustainability (Clewell 2000). Thus if the goal were to restore species or ecosystems characterized by cool, moist environments, then historical reference conditions could well be in conflict with future climate. This may be the case for relict populations of Chihuahua spruce, for example (Ledig et al. 2000). In contrast, in the specific case of the 2003 WFU fires at Grand Canyon, fire effects caused the ecosystems to move closer toward historical reference conditions while simultaneously reducing the living, dead, and ladder fuels that made the forest vulnerable to uncharacteristically severe fire. These changes help make the forests more resistant to the expected increases in fire size and severity (McKenzie et al. 2004).

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Table 1. Wildland fire use fires on the North Rim, Grand Canyon National Park, 2003. A total of 83 permanent monitoring plots established prior to the fires burned. An additional 25 and 18 permanent plots that did not burn were measured in comparison sites next to the low and high elevation fires, respectively. Fires and plots are mapped in Figure 1. Scientific names of species are shown in Table 2.

Fire Name	Date of Ignition	Final Size	No. Plots	Elevation of Plots	Forest Vegetation
Powell	June 15, 2003	1,460 ha (3,606 ac)	36	**Low**. 2,204 to 2,316 m (7,272 to 7,596 ft).	Ponderosa pine, Gambel oak, New Mexico locust
Big and Rose	August 24, 2003 (Big); October 7, 2003 (Rose)	1,592 ha (3,936 ac)	29	**Middle**. 2,433 to 2,560 m (7,980 to 8,397 ft).	Ponderosa pine, white fir, aspen
Poplar Complex	September 4, 2003	4,813 ha (11,889 ac)	18	**High**. 2,599 to 2,678 m (8,525 to 8,783 ft).	Subalpine fir, spruce, Douglas-fir, ponderosa pine, aspen

Table 2. Species names and codes used in the text.

Species	Common Name	Code
Abies lasiocarpa (Hook.) Nutt.	Subalpine fir	ABLA
Abies concolor (Gordon & Glendinning) Hoopes.	White fir	ABCO
Picea engelmannii Parry ex Engelm.	Engelmann spruce	PIEN
Picea pungens Engelm.	Blue spruce	Combined with PIEN
Pinus edulis Engelm.	Rocky Mountain pinyon	PIED
Pinus ponderosa var. *scopulorum* P. & C. Lawson	Ponderosa pine	PIPO
Populus tremuloides Michx.	Quaking aspen	POTR
Pseudotsuga menziesii (Mirb.) Franco var. *glauca* (Beissn.) Franco	Rocky Mountain Douglas-fir	PSME
Quercus gambelii Nutt.	Gambel oak	QUGA
Robinia neomexicana Gray	New Mexico locust	RONE

Table 3. Forest structure (trees ≥ 2.5 cm [1 inch] dbh) at low, middle, and high-elevation study sites before and after wildland fire use fires. Unburned comparison sites at low and high elevation are included. Species codes are shown in Table 2.

Metric Units

Site	Total	ABCO	ABLA	PIEN	PIED	PIPO	POTR	PSME	QUGA	RONE
					-- Tree Density (trees ha^{-1}) --					
Low Burn Pre-fire	574.7	0.0	0.0	0.0	0.0	217.1	0.0	0.0	276.0	81.5
Low Burn Post-fire	367.8	0.0	0.0	0.0	0.0	184.1	0.0	0.0	161.4	0.0
Percent change	*-36%*					*-15%*			*-42%*	*-100%*
Low Unburned Pre-fire	800.9	6.4	0.0	0.0	0.4	157.0	7.3	0.0	578.3	51.5
Low Unburned Post-fire	750.4	36.2	0.0	0.0	0.4	144.0	5.5	0.0	504.7	59.6
Percent change	*-6%*	*464%*			*0%*	*-8%*	*-25%*		*-13%*	*16%*
Mid Burn Pre-fire	930.4	459.6	0.0	0.0	0.0	156.3	256.0	52.8	0.0	5.8
Mid Burn Post-fire	428.5	208.8	0.0	0.0	0.0	107.2	89.0	20.7	0.0	0.0
Percent change	*-54%*	*-55%*				*-31%*	*-65%*	*-61%*		*-100%*
High Burn Pre-fire	1061.2	291.0	145.5	215.2	0.0	91.9	277.0	29.3	0.0	11.5
High Burn Post-fire	459.8	117.2	46.0	101.2	0.0	71.4	98.2	23.6	0.0	0.0
Percent change	*-57%*	*-60%*	*-68%*	*-53%*		*-22%*	*-65%*	*-19%*		*-100%*
High Unburned Pre-fire	1046.1	100.4	209.9	407.8	0.0	61.5	190.6	56.7	0.0	0.0
High Unburned Post-fire	930.6	88.2	202.0	366.9	0.0	60.7	156.1	54.5	0.0	0.0
Percent change	*-11%*	*-12%*	*-4%*	*-10%*		*-1%*	*-18%*	*-4%*		
					-- Basal Area (m^2 ha^{-1}) --					
Low Burn Pre-fire	26.2	0.0	0.0	0.0	0.0	24.3	0.0	0.0	1.6	0.3
Low Burn Post-fire	24.6	0.0	0.0	0.0	0.0	23.4	0.0	0.0	1.1	0.0
Percent change	*-6%*					*-4%*			*-35%*	*-100%*
Low Unburned Pre-fire	26.5	0.1	0.0	0.0	0.0	21.7	0.007	0.0	4.4	0.4
Low Unburned Post-fire	26.0	0.1	0.0	0.0	0.0	21.2	0.01	0.0	4.3	0.4
Percent change	*-2%*	*112%*			*8%*	*-2%*	*53%*		*-2%*	*6%*
Mid Burn Pre-fire	40.9	14.6	0.0	0.0	0.0	19.4	5.4	1.5	0.0	0.0
Mid Burn Post-fire	31.3	10.1	0.0	0.0	0.0	17.9	1.9	1.5	0.0	0.0
Percent change	*-23%*	*-31%*				*-8%*	*-65%*	*0%*		*-100%*
High Burn Pre-fire	34.7	10.7	2.3	6.5	0.0	7.1	6.3	1.9	0.0	0.0
High Burn Post-fire	20.3	4.1	0.4	4.0	0.0	6.7	3.1	1.9	0.0	0.0
Percent change	*-42%*	*-61%*	*-82%*	*-37%*		*-6%*	*-51%*	*2%*		*-100%*
High Unburned Pre-fire	33.3	4.7	4.4	9.4	0.0	4.5	6.3	4.0	0.0	0.0
High Unburned Post-fire	31.1	4.1	3.8	8.8	0.0	4.6	5.7	4.0	0.0	0.0
Percent change	*-6%*	*-11%*	*-12%*	*-6%*		*2%*	*-9%*	*0%*		

English Units

Site	Total	ABCO	ABLA	PIEN	PIED	PIPO	POTR	PSME	QUGA	RONE
					-- Tree Density (trees ac^{-1}) --					
Low Burn Pre-fire	232.6	0.0	0.0	0.0	0.0	87.9	0.0	0.0	111.7	33.0
Low Burn Post-fire	148.9	0.0	0.0	0.0	0.0	74.5	0.0	0.0	65.3	0.0
Percent change	*-36%*					*-15%*			*-42%*	*-100%*
Low Unburned Pre-fire	324.1	2.6	0.0	0.0	0.2	63.5	3.0	0.0	234.1	20.8
Low Unburned Post-fire	303.7	14.7	0.0	0.0	0.2	58.3	2.2	0.0	204.2	24.1
Percent change	*-6%*	*464%*			*0%*	*-8%*	*-25%*		*-13%*	*16%*
Mid Burn Pre-fire	376.5	186.0	0.0	0.0	0.0	63.2	103.6	21.4	0.0	2.3
Mid Burn Post-fire	173.4	84.5	0.0	0.0	0.0	43.4	36.0	8.4	0.0	0.0
Percent change	*-54%*	*-55%*				*-31%*	*-65%*	*-61%*		*-100%*
High Burn Pre-fire	429.5	117.8	58.9	87.1	0.0	37.2	112.1	11.8	0.0	4.6
High Burn Post-fire	186.1	47.4	18.6	40.9	0.0	28.9	39.7	9.6	0.0	0.0
Percent change	*-57%*	*-60%*	*-68%*	*-53%*		*-22%*	*-65%*	*-19%*		*-100%*
High Unburned Pre-fire	423.4	40.6	84.9	165.1	0.0	24.9	77.1	23.0	0.0	0.0
High Unburned Post-fire	376.6	35.7	81.7	148.5	0.0	24.6	63.2	22.0	0.0	0.0
Percent change	*-11%*	*-12%*	*-4%*	*-10%*		*-1%*	*-18%*	*-4%*		
					-- Basal Area (ft^2 ac^{-1}) --					
Low Burn Pre-fire	114.1	0.0	0.0	0.0	0.0	105.9	0.0	0.0	7.1	1.1
Low Burn Post-fire	106.9	0.0	0.0	0.0	0.0	102.0	0.0	0.0	4.6	0.0
Percent change	*-6%*					*-4%*			*-35%*	*-100%*
Low Unburned Pre-fire	115.6	0.3	0.0	0.0	0.0	94.4	0.0	0.0	19.2	1.6
Low Unburned Post-fire	113.4	0.5	0.0	0.0	0.0	92.3	0.0	0.0	18.8	1.7
Percent change	*-2%*	*112%*			*8%*	*-2%*	*53%*		*-2%*	*6%*
Mid Burn Pre-fire	177.9	63.5	0.0	0.0	0.0	84.5	23.3	6.5	0.0	0.0
Mid Burn Post-fire	136.4	43.8	0.0	0.0	0.0	77.9	8.2	6.6	0.0	0.0
Percent change	*-23%*	*-31%*				*-8%*	*-65%*	*0%*		*-100%*
High Burn Pre-fire	151.2	46.4	9.8	28.1	0.0	30.9	27.6	8.3	0.0	0.1
High Burn Post-fire	88.4	18.0	1.7	17.6	0.0	29.0	13.6	8.4	0.0	0.0
Percent change	*-42%*	*-61%*	*-82%*	*-37%*		*-6%*	*-51%*	*2%*		*-100%*
High Unburned Pre-fire	144.9	20.3	19.0	40.8	0.0	19.7	27.4	17.3	0.0	0.0
High Unburned Post-fire	135.7	18.0	16.6	38.5	0.0	20.1	24.8	17.3	0.0	0.0
Percent change	*-6%*	*-11%*	*-12%*	*-6%*	*0%*	*2%*	*-9%*	*0%*	*0%*	*0%*

Table 4. Canopy cover (percent) at low, middle, and high-elevation study sites before and after wildland fire use fires. Unburned comparison sites at low and high elevation are included. Canopy data were not measured on the High Unburned Post-fire plots.

	No. Plots	Mean	Minimum	Maximum	S.E.	Percent Change
LOW ELEVATION						
Low Burn Pre-fire	36	49.7%	15.4%	79.2%	2.1%	
Low Burn Post-fire	36	44.0%	12.5%	75.0%	2.6%	-10.0%
Low Unburned Pre-fire	25	48.3%	0.3%	85.5%	4.3%	
Low Unburned Post-fire	25	44.0%	0.0%	78.1%	3.5%	-8.9%
MIDDLE ELEVATION						
Mid Burn Pre-fire	29	63.2%	31.6%	84.9%	2.3%	
Mid Burn Post-fire	29	54.7%	28.1%	84.4%	3.0%	-13.4%
HIGH ELEVATION						
High Burn Pre-fire	18	52.5%	31.6%	84.6%	3.0%	
High Burn Post-fire	18	40.3%	21.9%	75.0%	3.6%	-23.4%
High Unburned Pre-fire	18	54.3%	22.9%	80.4%	3.4%	

Table 5. Mean Diameter and maximum bole char of surviving vs. killed overstory trees (dbh ≥ 15 cm) at low, middle, and high-elevation study sites after wildland fire use fires. Data are only presented for categories with ≥ 12 individuals. Significant differences (Mann-Whitney *U, P* < .05/3) between surviving and killed trees in each category are indicated with *.

Metric Units

Site	ABCO	ABLA	PIEN	PIPO	POTR	PSME
			Dbh (cm)			
Low Burn, Surviving				42.4*		
Low Burn, Killed				27.7*		
Middle Burn, Surviving	40.1*			41.1*	40.8*	45.1
Middle Burn, Killed	29.0*			31.3*	27.2*	
High Burn, Surviving	37.1*	42.4*	37.5*	39.1*	36.7	
High Burn, Killed	25.6*	30.0*	25.7*	30.3*	31.3	
			Bole char, maximum (m)			
Low Burn, Surviving				3.1*		
Low Burn, Killed				5.6*		
Middle Burn, Surviving	1.7*			2.4*	1.0*	1.9
Middle Burn, Killed	5.0*			9.2*	3.3*	
High Burn, Surviving	1.5*	1.9	1.7	2.4*	0.3*	
High Burn, Killed	2.6*	2.6	3.3	3.7*	4.2*	

English Units

Site	ABCO	ABLA	PIEN	PIPO	POTR	PSME
			Dbh (inch)			
Low Burn, Surviving				16.7*		
Low Burn, Killed				10.9*		
Middle Burn, Surviving	15.8*			16.2*	16.1*	17.8
Middle Burn, Killed	11.4*			12.3*	10.7*	
High Burn, Surviving	14.6*	16.7*	14.7	15.4*	14.4	
High Burn, Killed	10.1*	11.8*	10.1	11.9*	12.3	
			Bole char, maximum (ft)			
Low Burn, Surviving				10.1*		
Low Burn, Killed				18.5*		
Middle Burn, Surviving	5.6*			8.0*	3.3*	6.1
Middle Burn, Killed	16.4*			30.0*	11.0*	
High Burn, Surviving	4.9*	6.3	5.5	7.9*	1.0*	
High Burn, Killed	8.6*	8.5	10.8	12.1*	13.8*	

Table 6. Tree regeneration (seedlings and sprouts < 2.5 cm [1 inch] dbh) at low, middle, and high-elevation study sites before and after wildland fire use fires. Unburned comparison sites at low and high elevation are included. Species codes are shown in Table 2.

Metric Units

Site	Total	ABCO	ABLA	PIEN	PIPO	POTR	PSME	QUGA	RONE
					-- Tree Density (trees ha^{-1}) --				
Low Burn Pre-fire	5817.2	0.0	0.0	0.0	106.3	0.0	0.0	4050.0	1660.9
Low Burn Post-fire	8266.6	0.0	0.0	0.0	22.3	0.0	0.0	4916.8	3327.5
Percent change	*42%*				*-79%*			*21%*	*100%*
Low Unburned Pre-fire	6995.7	0.0	0.0	0.0	223.3	18.3	0.0	3479.5	3274.6
Low Unburned Post-fire	10223.7	62.9	0.0	0.0	470.5	9.2	0.0	6504.9	3176.3
Percent change	*46%*				*111%*	*-50%*		*87%*	*-3%*
Mid Burn Pre-fire	9112.2	4223.8	0.0	0.0	91.0	3726.0	48.8	0.0	1022.7
Mid Burn Post-fire	8855.6	6070.8	258.0	0.0	21.3	2152.5	28.5	0.0	324.5
Percent change	*-3%*	*44%*			*-77%*	*-42%*	*-42%*		*-68%*
High Burn Pre-fire	10061.1	1673.2	1962.1	970.9	247.1	4716.2	33.4	0.0	458.1
High Burn Post-fire	4998.4	1021.3	403.3	563.0	0.0	2954.4	33.4	0.0	22.9
Percent change	*-50%*	*-39%*	*-79%*	*-42%*	*-100%*	*-37%*	*0%*		*-95%*
High Unburned Pre-fire	8153.3	1308.5	2291.7	1301.1	22.6	2973.3	256.2	0.0	0.0
High Unburned Post-fire	6143.9	2320.2	1151.9	753.1	22.6	1736.9	159.3	0.0	0.0
Percent change	*-25%*	*77%*	*-50%*	*-42%*	*0%*	*-42%*	*-38%*		

English Units

Site	Total	ABCO	ABLA	PIEN	PIPO	POTR	PSME	QUGA	RONE
					-- Tree Density (trees ac^{-1}) --				
Low Burn Pre-fire	2354.2	0.0	0.0	0.0	43.0	0.0	0.0	1639.0	672.1
Low Burn Post-fire	3345.5	0.0	0.0	0.0	9.0	0.0	0.0	1989.8	1346.7
Percent change	*42%*				*-79%*			*21%*	*100%*
Low Unburned Pre-fire	2831.2	0.0	0.0	0.0	90.4	7.4	0.0	1408.1	1325.2
Low Unburned Post-fire	4137.5	25.4	0.0	0.0	190.4	3.7	0.0	2632.5	1285.4
Percent change	*46%*				*111%*	*-50%*		*87%*	*-3%*
Mid Burn Pre-fire	3687.7	1709.4	0.0	0.0	36.8	1507.9	19.7	0.0	413.9
Mid Burn Post-fire	3583.9	2456.8	104.4	0.0	8.6	871.1	11.5	0.0	131.3
Percent change	*-3%*	*44%*			*-77%*	*-42%*	*-42%*		*-68%*
High Burn Pre-fire	4071.7	677.1	794.1	392.9	100.0	1908.7	13.5	0.0	185.4
High Burn Post-fire	2022.9	413.3	163.2	227.9	0.0	1195.7	13.5	0.0	9.3
Percent change	*-50%*	*-39%*	*-79%*	*-42%*	*-100%*	*-37%*	*0%*		*-95%*
High Unburned Pre-fire	3299.6	529.5	927.5	526.6	9.1	1203.3	103.7	0.0	0.0
High Unburned Post-fire	2486.5	939.0	466.2	304.8	9.1	702.9	64.5	0.0	0.0
Percent change	*-25%*	*77%*	*-50%*	*-42%*	*0%*	*-42%*	*-38%*		

Table 7. Changes in forest floor and woody debris at low, middle, and high-elevation study sites before and after wildland fire use fires. Unburned comparison sites at low and high elevation are included. Fine woody debris is material with diameter < 7.62 cm (3 inches); coarse woody debris is larger material. S.E. = standard error of the mean.

Metric Units

	No. Plots	**Forest Floor Depth (cm)**	**S.E.**	**Fine Woody Debris (Mg ha^{-1})**	**S.E.**	**Coarse Woody Debris (Mg ha^{-1})**	**S.E.**
Low Burn Pre-fire	36	3.28	0.17	3.90	0.51	14.82	3.50
Low Burn Post-fire	36	1.86	0.09	2.53	0.31	8.91	2.13
Percent change		*-43.2%*		*-35.1%*		*-39.9%*	
Low Unburned Pre-fire	25	2.30	0.21	3.30	0.45	14.92	5.54
Low Unburned Post-fire	25	2.27	0.18	3.90	0.51	20.19	5.93
Percent change		*-1.5%*		*61.8%*		*35.3%*	
Mid Burn Pre-fire	29	3.95	0.29	13.20	1.26	53.89	6.40
Mid Burn Post-fire	29	1.22	0.09	6.49	0.68	33.24	9.94
Percent change		*-69.0%*		*-50.8%*		*-38.3%*	
High Burn Pre-fire	18	3.75	0.33	12.02	1.14	56.42	14.34
High Burn Post-fire	18	1.34	0.18	7.23	1.09	22.85	4.72
Percent change		*-64.1%*		*-39.9%*		*-59.5%*	
High Unburned Pre-fire	18	3.26	0.25	12.34	1.42	36.96	5.27
High Unburned Post-fire	18	2.71	0.26	9.85	1.07	46.14	9.52
Percent change		*-16.7%*		*-20.2%*		*24.8%*	

English Units

	No. Plots	**Forest Floor Depth (inch)**	**S.E.**	**Fine Woody Debris (ton ac^{-1})**	**S.E.**	**Coarse Woody Debris (ton ac^{-1})**	**S.E.**
Low Burn Pre-fire	36	1.29	0.07	1.74	0.23	6.61	1.56
Low Burn Post-fire	36	0.73	0.03	1.13	0.14	3.97	0.95
Percent change		*-43.2%*		*-35.1%*		*-39.9%*	
Low Unburned Pre-fire	25	0.91	0.08	1.47	0.20	6.65	2.47
Low Unburned Post-fire	25	0.89	0.07	1.74	0.23	9.01	2.65
Percent change		*-1.5%*		*61.8%*		*35.3%*	
Mid Burn Pre-fire	29	1.56	0.11	5.89	0.56	24.04	2.86
Mid Burn Post-fire	29	0.48	0.03	2.90	0.31	14.83	4.44
Percent change		*-69.0%*		*-50.8%*		*-38.3%*	
High Burn Pre-fire	18	1.47	0.13	5.36	0.51	25.17	6.40
High Burn Post-fire	18	0.53	0.07	3.23	0.48	10.19	2.11
Percent change		*-64.1%*		*-39.9%*		*-59.5%*	
High Unburned Pre-fire	18	1.28	0.10	5.51	0.63	16.49	2.35
High Unburned Post-fire	18	1.07	0.10	4.40	0.48	20.59	4.25
Percent change		*-16.7%*		*-20.2%*		*24.8%*	

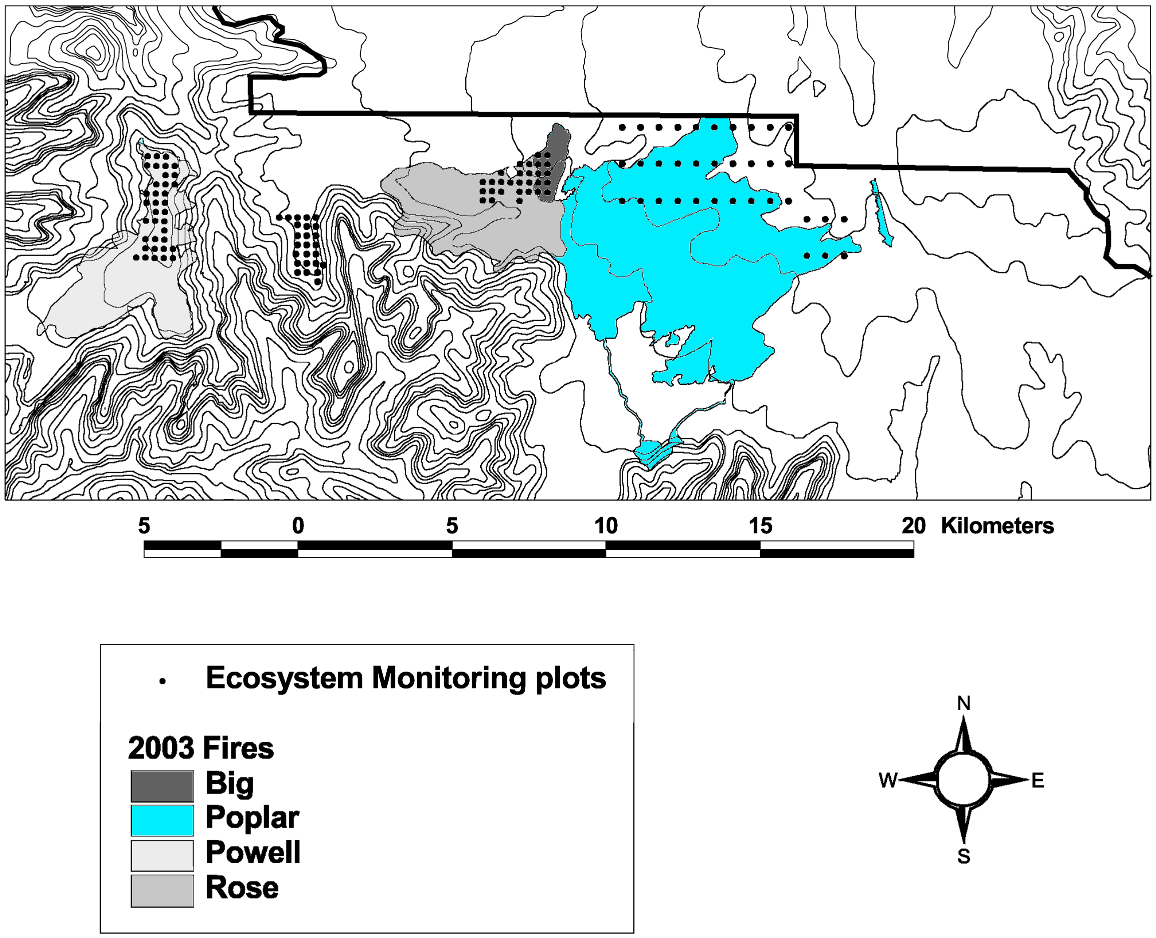


Figure 1. Wildland fire use fires burning in 2003 (shaded polygons) are mapped over grids of permanent plots established between 1997-2001 on the North Rim. The boundary line separates Grand Canyon National Park (south) from Kaibab National Forest (north). From west to east, the study areas are the low-elevation burned site (Powell Plateau), the low-elevation unburned site (Rainbow Plateau), the middle-elevation burned site (Swamp Ridge), and the high-elevation burned and unburned sites (Big Spring and Little Park).

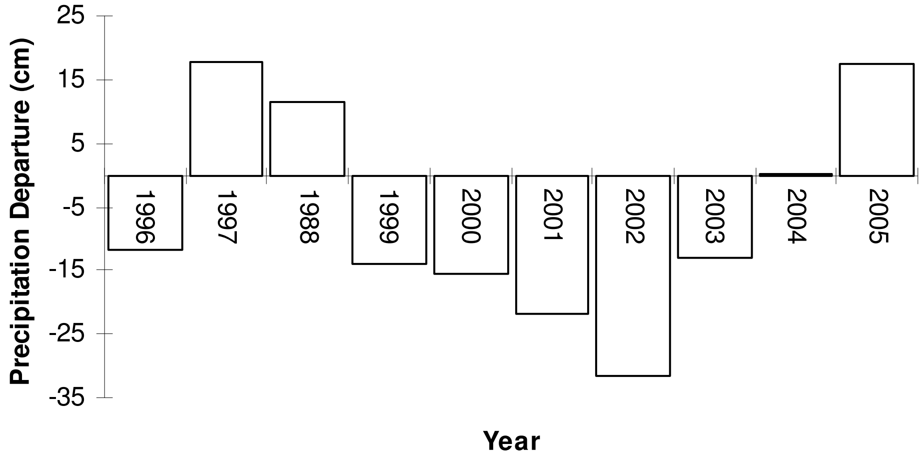


Figure 2. Precipitation departure (cm) from the 1948-2005 average of 65.8 cm during the decade 1996-2005. Below-average precipitation occurred in 6 of the 10 years. The year 2002 was the driest on record, recording only 53% of average. Records were missing in some years, especially winters of 1959-1973.

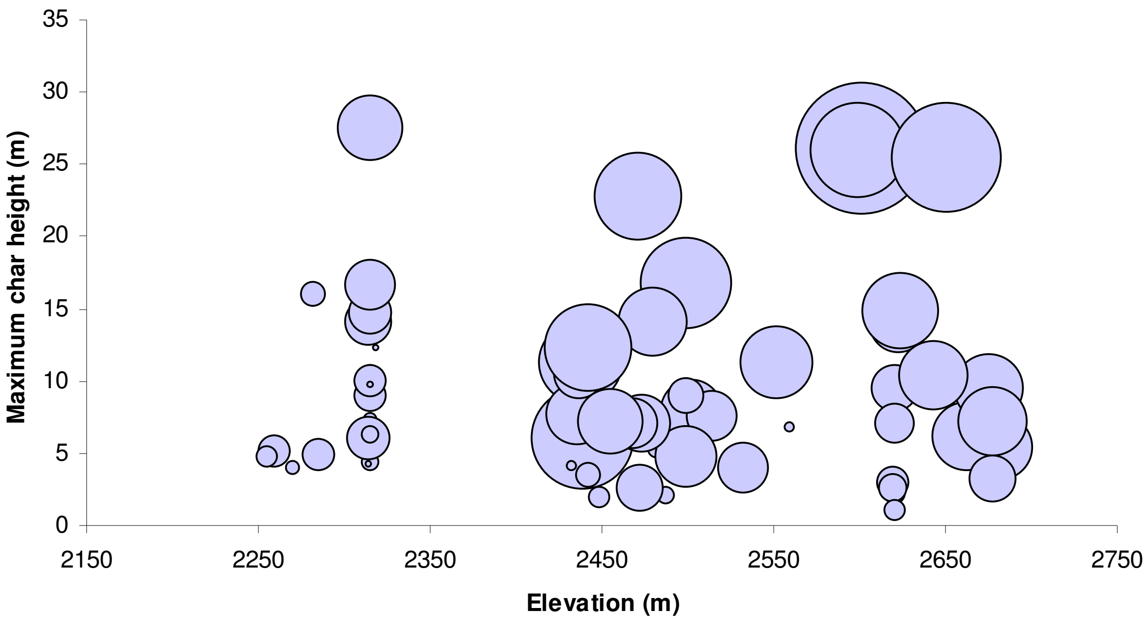


Figure 3. Distribution of maximum char height (highest char value per plot) as an indicator of fire intensity over all the burned plots from lowest to highest elevation. Bubbles are proportional to basal area mortality between the pre- and post-fire measurements.

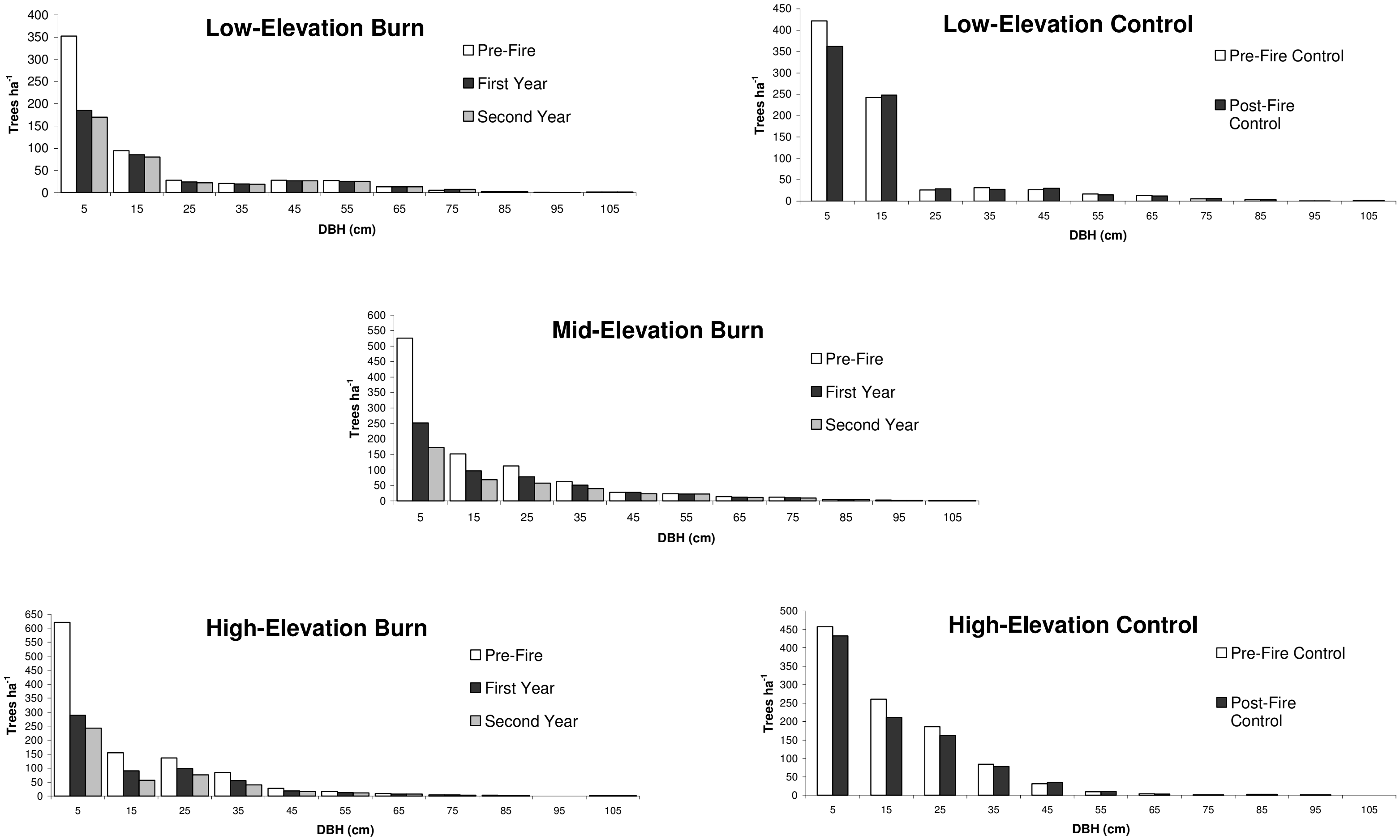
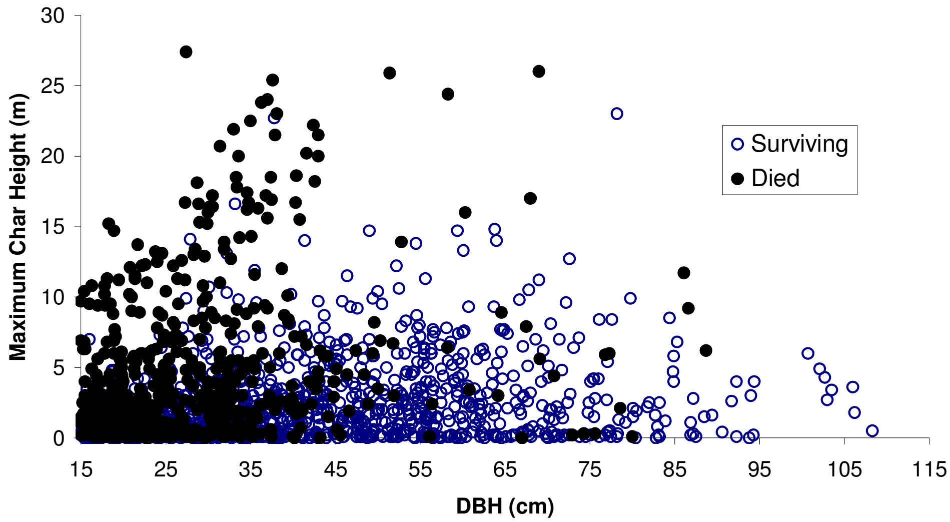


Figure 4. Changes in diameter distribution at sites burned in Wildland Fire Use fires over an elevational gradient. Diameter class midpoints are shown on the x-axes; minimum diameter is 2.5 cm. Tree survival is shown in both the first year after fire (2004) and the second year (2005). Unburned comparison sites were measured once, either in 2004 or 2005.



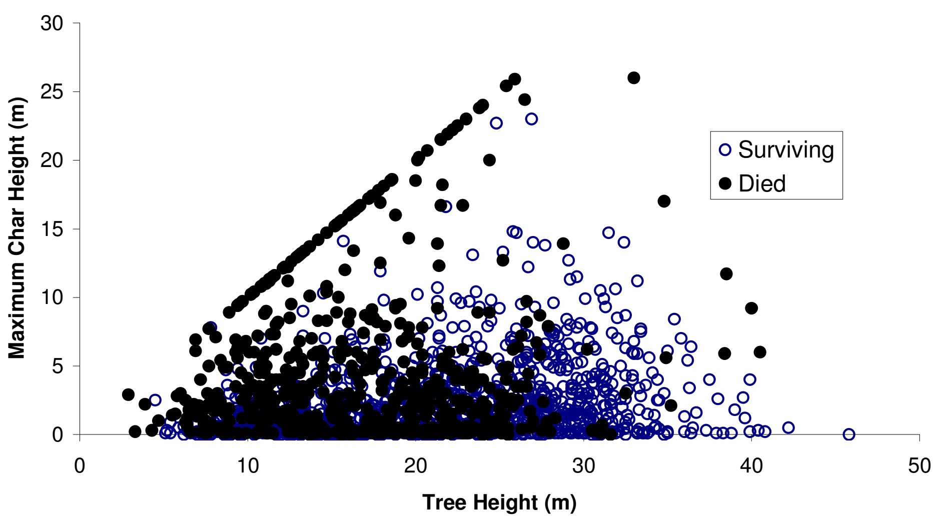


Figure 5. Maximum char height, as a measure of heat affecting the tree, compared with dbh and tree height. Smaller and shorter trees, especially those with high char, were most likely to die.

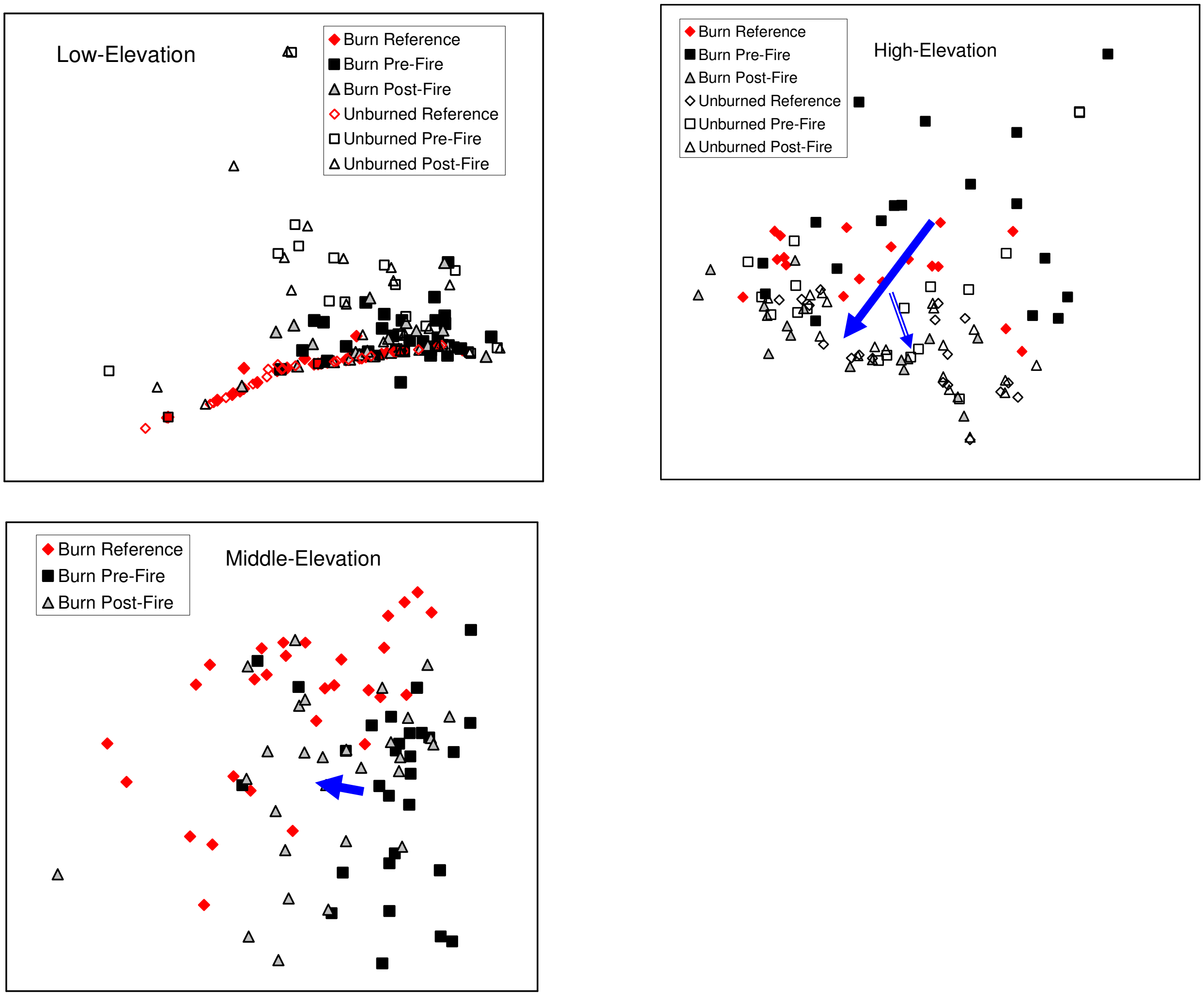


Figure 6. Non-metric multidimensional scaling ordination of basal area of all species reconstructed at the onset of fire exclusion (reference condition, 1880), before WFU, and after WFU. Changes in the centroid of each distribution from before to after the WFU dates are shown with solid arrows (burned sites) or open arrows (unburned sites). Changes at the low-elevation site were too minor to be graphed.